Int J Life Cycle Assess (2012) 17:795–812
DOI 10.1007/s11367-012-0403-1

INPUT-OUTPUT AND HYBRID LCA

An input–output based framework to evaluate human labour in life cycle assessment

Benedetto Rugani · Daryna Panasiuk · Enrico Benetto

Received: 21 September 2011 / Accepted: 27 February 2012 / Published online: 15 March 2012

Abstract
Purpose In LCA, the intrinsic dependence of productions to human labour (HL) activities is usually neglected, without providing any clear arguments. HL is not considered to be related to and affected by, changes to the functional unit, although this is evidently not the case. This research aims at investigating the relationship between HL and LCA and at developing an operational framework to assess the life cycle environmental impact of HL in LCA.
Materials and methods System boundaries and functional unit (euro/working-hour) of HL were defined. Statistical datasets of household expenditures (HEs) allowed differentiating among human consumption behaviours and the definition of three HL types, based on different work skills: HL-1 (qualified worker), HL-2 (technician), and HL-3 (manual worker). The HEs framework of Luxembourg was used because of data availability and was then extended to other EU-27 countries. A comparative LCIA of the HLs types was carried out using an environmentally extended input–output model (EU-27). Afterwards, ten agri-food and industrial LCAs case studies were modified for hybrid LCAs, adding HL input to LCIs and using the ReCiPe midpoint method for LCIA.
Results and discussion The LCIA comparison of HLs shows that HL-1 generates environmental impacts that are always greater than HL-2 and HL-3, e.g. 1 h of HL-1, which involves workers with the highest consumption of goods and services, does generate 0.52 kg CO_2-eq, whereas HL-2 and HL-3 generate 0.46 and 0.41 kg CO_2-eq, respectively. The impact of average HL is higher in EU countries with the highest HEs budgets, e.g. the average HL impact in Luxembourg is 28% to 79% higher than the corresponding HL impact in other EU-27 countries. Within the case studies, the HL significantly contributes to the total impact for several categories (e.g. fossil and ozone depletion up to 16% and 20%, respectively). Despite these important results, some limitations due to data and models used are investigated to suggest further methodological improvements.
Conclusions The integration of HL inputs to product LCIs can improve accuracy of the entire life cycle analysis, since no product would exist without direct and/or indirect HL. This leads considering humans at the same level of technological/economic activities that cause environmental damage, with humans being perceived as leading actors and explicitly responsible for the impacts. What remains an open question is how to account for non-physical information, such as knowledge/education/culture, which distinguish humans from machinery and are essential items for our future sustainable development.

Keywords Environmentally Extended Input–Output Table (EEIOT) · Household consumption · Human labour · Hybrid LCA · Input–output LCA · Process-LCA · Purchased power standard (PPS) · Working hour

Responsible editor: Shinichiro Nakamura

Electronic supplementary material The online version of this article (doi:10.1007/s11367-012-0403-1) contains supplementary material, which is available to authorized users.

B. Rugani (✉) · D. Panasiuk · E. Benetto
Public Research Centre Henri Tudor (CRPHT)/Resource Centre for Environmental Technologies (CRTE),
66, rue de Luxembourg, P.B. 144, 4002, Esch-sur-Alzette, Luxembourg
e-mail: benedetto.rugani@tudor.lu

1 Introduction

Many ecosystems are dominated directly by humanity, and no ecosystem on the Earth's surface is free of pervasive

human influences (Vitousek et al. 1997). Humans are responsible for climate change, for the depletion of natural resources, biodiversity and the ozone layer, as well as for damage to human health and ecosystems. Despite this evidence, the contribution of human activities and in particular of human labour (HL) is not usually included in the environmental assessment of products and processes. Therefore, the question is: how could the contribution of HL to the environmental assessment of products be evaluated?

Life Cycle Assessment (LCA) holds a utilitarian and anthropocentric perspective. The basic assumption is that only activities which are related to and affected by changes of the functional unit are included in the Life Cycle Inventory (LCI) and finally in the Life Cycle Impact Assessment (LCIA). Historically this assumption led to the inclusion in LCI of only those activities which are directly or indirectly linked to the functional unit by exchanges of commodities, i.e. co-products and raw materials. However, during the last decade, the boundaries of LCI have progressively been enlarged to include activities which are linked to the functional unit by market driven mechanisms (Suh and Huppes 2002; Weidema 2006). If the inclusion of market driven activities is straightforward in a consequential LCI approach, it is generally not addressed in attributional LCIs. This is probably why the contribution of human labour is normally not included in LCIA (Spirinckx and Ceuterick 1996; Milà et al. 1998; Nebel et al. 2006). Nevertheless, from our point of view, this is an inconsistency, since human activities and specifically HL, fully support the operation of products life cycles and are, in turn, affected by changes of the functional unit. If a direct link between the use of resources and production systems does exist, as it is normally modeled in LCIs, production systems are also connected to and sustained by human life, which in turn consumes resources and releases emissions for its well-being. An intrinsic relation among biosphere, technosphere, and human life cycle exists and is tied together by the exchange of material and energy flows. Therefore, taking into account HL in attributional LCI (and in the related LCIA) leads to the further question: to what extend does human activity and in particular human labour, actually contribute to the overall environmental impact of a product?

This work aims at proposing a methodological framework to evaluate the contribution of human labour to the LCA. This paper does not deal with the evaluation of human labour in Social Life Cycle Assessment (SLCA), e.g. impact categories representing the labour rights (Dreyer et al. 2010; Jørgensen et al. 2010). SLCA is of great significance when looking at the global performance of a product and at its consequences for human well-being, but has a different scope and meaning when compared to a conventional Environmental LCA. Also, the aim was not to develop a framework to account for HL in LCI. We propose instead a methodology that can be used in LCA to assess the human labour supporting the life cycle of a product or process. Comprehensive studies evaluating the impact of the "lifecycle of human life and labour" have not been found in the literature. A few publications are available but these evaluate only portions of the complete life cycle (e.g. food supply, human activity wastes; Muñoz et al. 2008, 2010). Recently, a sensitivity analysis performed on fuel ethanol production chains, showed that the inclusion of HL, calculated with energy intensity factors (Nguyen et al. 2007) does not significantly affect the LCA results (Nguyen and Gheewala 2008a,b; Silalertruksa and Gheewala 2009). However, the authors highlight the need to develop a consistent method to account for HL in LCA.

Outside of LCA, other methods (such as energy-based analyses, extended exergy analysis) have explicitly tackled the evaluation of HL, mostly based on energetic assessments of metabolic consumption (Giampietro and Pimentel 1990; Fluck 1992; Giampietro 1997; Sciubba and Ulgiati 2005; Sciubba et al. 2008; Giampietro et al. 2006). Several authors highlighted the need to compare the HL environmental impact to those of machinery and investigated, by replacing machinery with labour, how sustainable manufacturing could be supported (Fluck 1981; Pimentel 1993; Bender 2003; Krausmann 2004; Zhang and Dornfeld 2007; Guzmán and Alonso 2008). To this end, the energy embodied and/or consumed in HL has been evaluated, in particular for organic agricultural production, since in most cases, this requires higher labour inputs than for conventional agriculture (Sorrells and Pimentel 1981; Loake 2001; Guzmán and Alonso 2008). When assessing the sustainability of society's metabolism, attention has been paid to the incorporation of energy from food intake necessary for HL (Haberl 2001a, b). This incorporation was observed to be of negligible impact to the global energy balance of modern societies, when compared to machine power or other technical device energies (Haberl et al. 2006; Krausmann and Haberl 2002). The eMergy evaluation method (Odum 1988; Odum 1996) considers accounting for HL an integral part of the human-dominated systems analysis. In eMergy, the HL is usually quantified by multiplying its cost by the average eMergy to money ratio of an economy (i.e. ratio of eMergy of a country or a region to its GDP), or by dividing the annual amount of country/region eMergy by the number of people in the country/region, multiplied by their metabolic energy requirement (Campbell 1998; Pulselli 2010). However, an intense debate is ongoing within the emergy community whether to calculate the eMergy of people and labour on the base of educational attainment or by considering people at the same level, estimating the eMergy behind any expenditure for services purchased by humans (Abel 2010, Brown and Herendeen 1996, Odum 1988). Moreover, the eMergy procedure suffers from lack of completeness and accuracy of the background inventory (Hau and Bakshi 2004; Ingwersen 2010), which make roughly approximated the calculation for human labour.

Much data concerning human activity is available in statistical and economic databases. However, information contained in these national and international sources is mainly used to derive social and economic impact indicators (e.g. classification of expenditures for human consumption sectors to calculate poverty indicators among different classes within nations; Eurostat 2008a) and therefore some adjustment is required for their use in environmental assessment. In this work, we adopt an extended input–output based model to gather environmental data and assess both direct and indirect impacts caused by HL. Input–output tables with environmental extensions supply environmental information on economic activities based on a relatively complete system, while requiring little time and resources (Suh and Huppes 2002). Using this framework, we could perform an IO-LCA hybrid analysis, which is aimed at enlarging the evaluation perspective of human contribution in LCA.

Table 1 Midpoint impact categories used for LCIA of human labour and related case studies: Recipe method (Europe H) (Goedkoop et al. 2009)

Impact category	Abbreviation	Unit
Climate change	CC	kg CO_2-eq
Ozone depletion	OD	kg CFC-11 eq
Human toxicity	HT	kg 1,4-DB eq
Photochemical oxidant formation	POF	kg NMVOC
Particulate matter formation	PMF	kg PM10 eq
Terrestrial acidification	TA	kg SO_2 eq
Freshwater eutrophication	FE	kg P eq
Marine eutrophication	ME	kg N eq
Terrestrial ecotoxicity	TET	kg 1,4-DB eq
Freshwater ecotoxicity	FET	kg 1,4-DB eq
Marine ecotoxicity	MET	kg 1,4-DB eq
Agricultural land occupation	ALO	m^2·year
Urban land occupation	ULO	m^2·year
Metal depletion	MRD	kg Fe eq
Fossil depletion	FD	kg oil eq

2 Outline of the research work

The assessment of HL contribution in LCA has been structured into three steps:

i) The definition of the methodology to account for different HL types
ii) The LCIA and the comparison of the different HL types
iii) Testing of the overall framework by means of case studies

In step (i), system boundaries and the functional unit of HL have been defined, as described in Section 3.1. Three types of HLs are placed within the framework of the EU-27 countries, by using statistical datasets based on household expenditures per economic sector (see Section 3.2). Subsequently, in step (ii) a comparative LCIA for the selected HL types was carried out (Section 4) using an environmentally extended input–output model, covering EU-27 countries, as described in Section 3.3. Finally, in step (iii), a number of hybrid LCAs are considered where HL has been implemented as a product system for LCA models and its contribution assessed within ten case studies (Section 5). The LCA calculations were performed using SimaPro. For LCIA, the ReCiPe method (Goedkoop et al. 2009) was used (Table 1).

3 Step I: methodological framework

3.1 System boundary definition and functional unit of the human labour's product system

Human labour can be considered as part of a broader "human life cycle system", covering the entire lifetime of a person, which includes all of that person's activities and all of the past and current contributions supporting these activities. We consider human labour to be directly dependent on the economic market and therefore linked to a certain number of physical processes (e.g. production of goods and services) that support it. However, a meaningful portion of processes which support HL is inherently outside economic markets, being of a cultural (non-physical) nature. For instance, the information and knowledge that makes a human being a "system able to perform a specific labour" is certainly important, but very difficult to evaluate since it is related to less tangible processes that, to further complicate things, mainly occurred in the past. In this preliminary approach, due to the complexity of the issue, the non-physical and non-conventional direct contributions are excluded (Fig. 1). For example, we do not make provisions for the inclusion of genetic information or educational experiences in the model.

We recognize that person grow up and during this time gains their knowledge within their particular household framework. Indeed, an average household framework can be interpreted as the core system for workers to maintain their well-being living standards. Worker's expenditures are also necessary for their children education (household members who do not work) up to the time they are able to be independent (at the beginning of their labour activity). Furthermore, household expenditures (HEs) include expenditures incurred on the domestic territory (by residents and non-residents) for the direct fulfillment of individual needs and cover the purchase of goods and services, which includes the consumption of own production (e.g. garden produce) and the rent of owner-occupied dwellings (Eurostat 2008a). Therefore, the educational and knowledge pool (e.g. culture, past work and travel experiences, communication capabilities) supporting human labour can

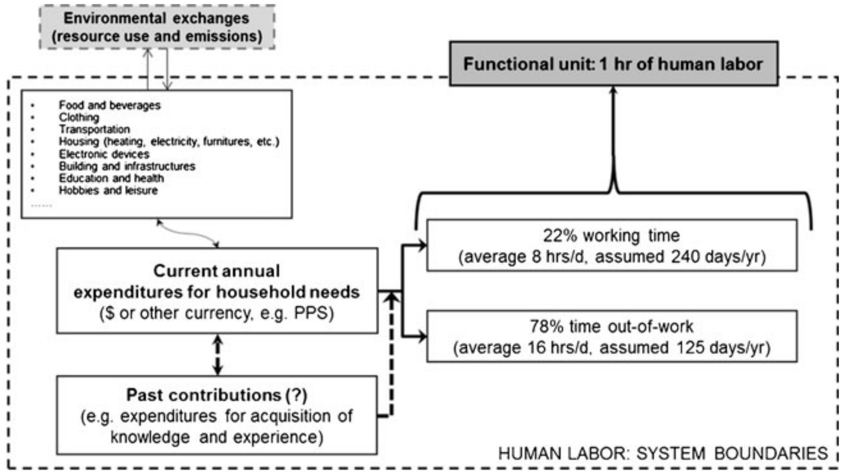

Fig. 1 System boundaries of human labour's product system; past contributions are conceptually included but explicitly not accounted for

be assumed to be dependent on the inputs (energy and materials) consumed to produce the goods and services required during the entire life of the worker within its household framework (see Fig. 1). In order to include the past contributions, the current household money invested in the well-being of not-working members (children and/or teenagers) could be added to the current expenditures of working people. This would be a first approximation, based on the assumption that working members in their past childhood needed similar investments into their education and well-being as today's children and teenagers. However, this assumption suffers from a severe limitation, because the household consumption behaviours and the technologies used to produce goods and services have changed over the decades and generations. Because of the lack of information to assign past contributions to the present human labour, we eventually decided not to account for the inputs of knowledge, culture, and expertise. Only processes that are directly related to the human living environment have been assessed. For example, the goods and services that usually ensure both primary and non-primary needs to the household working members (e.g. household purchases of food and beverages, heating and electricity, transportation, clothes, water for sanitary uses, health care, infrastructures, leisure purposes, cultural activities) have been included in the system boundaries. In contrast, all the life cycle processes related to services, infrastructures, tools, or other materials that directly relate to the working environment of the person and used during their labour have been excluded from the calculation (e.g. an office building, computers, work related transport, a mill, agricultural field, farm, tractor). Indeed, the contribution of these production/infrastructure/service processes actually belongs to the life cycle of the produced goods or services rather than to support HL. The processes included in the HL system boundary have been selected among the economic sectors producing goods and services annually purchased by households for their maintenance (see Section 3.2).

With regards to the functional unit of the HL's product system, the environmental inputs and outputs occurring during the whole year (8,760 h/year) were considered at first. This includes not only working hours, but also hours dedicated to eat and sleep, for hobbies and so forth, which are necessary to sustain human labour. Similarly, week-ends and days off work, as well as holiday periods, should be included in the system boundaries according to the rationale that all these activities enable people to recover and then be able to work again. We assumed an average of 240 working days per year with 8 working hours per day, which means that 22% of the overall annual time is dedicated to labour (1,920 h/year).

Therefore, 22% of an average life hour (and related expenditures) is allocated to human labour. In this manner, we were able to spread the environmental burdens that occur during "not-working" hours of the day (when eating or when travelling to work), but which are of essence for HL.

3.2 Human labour types

Relevant literature in the field of sustainable consumption acknowledges the evaluation of "household expenditures" as the best way to account for material and energy use patterns, with household consumptions being the "mirror" of the population lifestyle (Hertwich 2005, 2011; Druckman et al. 2008; Druckman and Jackson 2009; Moll et al. 2005). Accordingly, there is a vast literature on the environmental impact assessment of household expenditure patterns (among which Lenzen and Peters 2010; Munksgaard et al. 2005; Nijdam et al. 2005; Wier et al. 2001), where different kinds of variations have been explored: income, age, household types, etc. However, there is no study which explicitly highlights the inherent linkage between household consumption expenditures and labour, at least with connection to LCA. Indeed, the HE refers to any spending done by a person living alone, or by a group of people living together in shared accommodation with common domestic expenses (Eurostat 2008a). HEs are therefore used to distinguish among different types of human labour. We used the concept of adult equivalent (AD), which is an aggregate indicator for a household size,

adjusted to the number of persons in the household. That number is based on the amount of food/resources each person consumes within the household at country level (Eurostat-OECD 2006). AD scales, compared to per capita measures provides information on the contribution of various household members to the consumption behaviour of the household (Tedford et al. 1986). Household size is measured in AD terms to allow comparison of households of different sizes and structures, where the equivalence scale is the following: a weight of 1 is assigned to the household head, a weight of 0.5 to each additional adult, and a weight of 0.3 for each child aged 14 or younger (Da Costa et al. 2011). These scales have shown to be useful tools for narrowing the difference between the per capita estimates and real food consumption and enable the comparison of data for families with different compositions (Claro et al. 2010).

Three types of HL are defined in Table 2 on the basis of HEs per adult equivalent: (a) directors, seniors, qualified and intellectual workers (HL-1), technicians and clerks (HL-2), and manual workers (HL-3). These three types are founded on average HEs from three different groups of workers as outlined by STATEC Luxembourg (2008). Table 2 reports values both in Euro (reference year, 2005) and in percentage contribution of expenditures, along with a number of economic sectors that produce goods and services. The economic sectors in Table 2 correspond to those listed in the input–output table EU-27, and refer to the NACE classification (Eurostat 2008b; see Section 3.3). Since the original data of household expenditures refer to the COICOP (3-digits) commodities classification (UN Statistics Division 2011), we have adapted the flows according to the NACE classification method (see Electronic Supplementary Material, Table S1).

Data from STATEC (2008) in Table 2 originally referred to the entire household size. In order to convert those values in AD we applied the following formula:

$$\mathrm{Ea}_{\mathrm{EURO}} = \frac{\mathrm{Ea}_{\mathrm{PPS}} \cdot \mathrm{Eh}_{\mathrm{EURO}}}{\mathrm{Eh}_{\mathrm{PPS}}} \qquad (1)$$

where $\mathrm{Ea}_{\mathrm{EURO}}$ is the average HE per adult equivalent (in Euro) used to differentiate among HL types, $\mathrm{Ea}_{\mathrm{PPS}}$ is the average HE per adult equivalent in PPS (available from the Eurostat database Eurostat 2011), $\mathrm{Eh}_{\mathrm{EURO}}$ is the average HE per household in euro (available from the STATEC database; STATEC 2008), and $\mathrm{Eh}_{\mathrm{PPS}}$ is the average HE per household in PPS (available from the Eurostat database; Eurostat 2011).

Average HE per household and/or per AD among different countries are built through the use of comparative price levels, which represent the ratio between Purchasing Power Parities (PPPs) and market exchange rate for each country. PPPs are currency conversion rates that convert economic indicators expressed in national currencies into a common currency, called Purchasing Power Standard (PPS; Eurostat-OECD 2006). This is a kind of artificial common currency (i.e. PPS), which allows eliminating price level differences between countries or regions and calculating the real purchasing power of the resident population (Eurostat-OECD 2006). The ratio is shown in relation to the EU average (i.e. EU27=100): if the index of the comparative price levels for a country is higher/lower than 100, the country concerned is more expensive/cheap as compared to the EU average (Eurostat-OECD 2006). Prices of a basket of comparable and representative goods and services were collected for all considered countries in order to compile PPPs. For consumer products, about 3,000 products are priced and then aggregated into a limited number of categories (see Table 2). In addition, prices for housing, investment goods (including construction services), and government services were collected (STATEC 2008). The expenditure expressed in PPS (or "real expenditure") refers to an expenditure aggregate (e.g. GDP or actual individual consumption), converted to a common technical currency and a common price level using PPPs, which results in a set of comparable data across countries. For the purpose of international comparisons, if the real expenditure on, for instance, GDP is divided by the number of inhabitants in each country, the resulting real expenditure per inhabitant can be used as an indicator of the relative standard of living of the inhabitants of each country (Eurostat 2008a). Therefore, HEs expressed in PPS are used in this study for consistent spatial comparisons among different countries. Table 3 lists the typical annual HE per adult equivalent in PPS for the EU-27 countries and the relative PPP factor, as well as the functional units (in PPS/h) used for HL's comparisons among countries.

Values in Table 3 refer to the year 2005 and have been collected from Eurostat database sources (Eurostat 2011). These statistical sources also include the detailed HE per economic sector at COICOP level (UN Statistics Division 2011). Detailed COICOP statistics of HEs for all the considered EU countries are available only by household and not per AD. The framework of Luxembourg could be possibly adapted to the other European countries to differentiate among labours. Table 2 shows the percentage of variation from the average HE of each HL type, calculated for all the economic sectors. One could assume that the variations among labour types observed in Luxembourg are pertinent also for all the other European countries and therefore apply these differences to the average HEs per AD of Table 3. However, a comparison of LCIA results for different countries was performed only for one generic type of labour (identified by the AD framework in Table 3). The percent distribution of HEs from COICOP tables by household was adapted to the AD framework and then expenditures adjusted to the NACE classification (see the Electronic Supplementary Material, Tables S1 and S2).

Figure 2 shows the HE of the AD in Europe (reference year: 2005), which mostly focuses its expenditures (≈55%)

Table 2 Average annual HEs per adult equivalent (in Euro) for the three types of human labour in Luxembourg; functional units (Euro/h) are calculated considering an allocation factor of 22% (see Section 3.1)

35 Economic sectors of household expenditure (NACE classification)	Directors, seniors, qualified and intellectual workers (HL-1)		Technicians, clerks (HL-2)		Manual workers (HL-3)		Average, mean HE (adult eq.; 1.03)[a]		HL-1 (1.20)[a]	HL-2 (1.02)[a]	HL-3 (0.87)[a]
	€		€		€		€		Departure from the mean		
Air transport	463.19	0.97%	242.10	0.60%	124.83	0.36%	276.71	0.67%	67.39%	−12.51%	−54.89%
Coal, lignite, peat	11.18	0.02%	20.33	0.05%	23.00	0.07%	18.17	0.04%	−38.49%	11.90%	26.60%
Computer and related services	8.96	0.02%	5.56	0.01%	4.14	0.01%	6.22	0.02%	44.02%	−10.56%	−33.46%
Crude petroleum and natural gas	971.27	2.03%	1,010.09	2.49%	971.37	2.79%	984.24	2.40%	−1.32%	2.63%	−1.31%
Education	358.92	0.75%	204.87	0.50%	98.93	0.28%	220.91	0.54%	62.47%	−7.26%	−55.22%
Electrical machinery	274.78	0.57%	202.05	0.50%	156.80	0.45%	211.21	0.51%	30.10%	−4.34%	−25.76%
Electricity	424.56	0.89%	466.92	1.15%	494.52	1.42%	462.00	1.12%	−8.10%	1.07%	7.04%
Financial intermediation	234.66	0.49%	283.82	0.70%	210.92	0.61%	243.13	0.59%	−3.49%	16.74%	−13.25%
Food product and beverage	4011.68	8.39%	3589.01	8.84%	3626.68	10.41%	3742.45	9.11%	7.19%	−4.10%	−3.09%
Furniture and other manufactured goods	3,097.44	6.48%	2,261.38	5.57%	1,617.68	4.64%	2,325.50	5.66%	33.19%	−2.76%	−30.44%
Health and social work	384.27	0.80%	314.43	0.77%	319.14	0.92%	339.28	0.83%	13.26%	−7.32%	−5.94%
Hotels and restaurants	5,211.19	10.90%	3267.19	8.05%	2126.20	6.10%	3534.86	8.60%	47.42%	−7.57%	−39.85%
Instruments, medical, precision, optical, clocks	478.85	1.00%	449.32	1.11%	307.29	0.88%	411.82	1.00%	16.28%	9.11%	−25.38%
Insurance and pension funding	1,360.27	2.84%	1,382.69	3.40%	1,289.40	3.70%	1344.12	3.27%	1.20%	2.87%	−4.07%
Land transport	206.14	0.43%	114.05	0.28%	87.48	0.25%	135.89	0.33%	51.70%	−16.07%	−35.63%
Leather products, footwear	606.62	1.27%	501.70	1.24%	459.47	1.32%	522.60	1.27%	16.08%	−4.00%	−12.08%
Machinery and equipment	693.49	1.45%	597.55	1.47%	568.60	1.63%	619.88	1.51%	11.87%	−3.60%	−8.27%
Manuf. of chemicals, chemical prod. and man-made fibres	900.45	1.88%	830.86	2.05%	900.23	2.58%	877.18	2.13%	2.65%	−5.28%	2.63%
Manufacture of motor vehicles, trailers	2,766.01	5.78%	3,639.17	8.96%	3,181.10	9.13%	3195.43	7.78%	−13.44%	13.89%	−0.45%
Post and telecommunication	823.74	1.72%	748.72	1.84%	735.12	2.11%	769.19	1.87%	7.09%	−2.66%	−4.43%
Printed matter	456.42	0.95%	332.98	0.82%	237.44	0.68%	342.28	0.83%	33.35%	−2.72%	−30.63%
Public service and security	505.35	1.06%	225.18	0.55%	175.66	0.50%	302.06	0.74%	67.30%	−25.45%	−41.85%
Radio, television, communication equipment	351.51	0.74%	314.30	0.77%	241.43	0.69%	302.41	0.74%	16.23%	3.93%	−20.17%
Real estate services	12,125.60	25.35%	10459.66	25.76%	9528.55	27.35%	10,704.60	26.05%	13.27%	−2.29%	−10.99%
Recreation and culture	1,735.97	3.63%	1,331.77	3.28%	976.67	2.80%	1,348.14	3.28%	28.77%	−1.21%	−27.55%
Retail trade and repair services	602.99	1.26%	478.69	1.18%	381.39	1.09%	487.69	1.19%	23.64%	−1.85%	−21.80%
Sanitation, sewage, disposal	190.91	0.40%	202.23	0.50%	194.60	0.56%	195.92	0.48%	−2.55%	3.22%	−0.67%
Services n.e.c.	2,380.59	4.98%	1529.75	3.77%	1,041.83	2.99%	1650.72	4.02%	44.21%	−7.33%	−36.89%
Textile (household)	270.56	0.57%	272.40	0.67%	190.18	0.55%	244.38	0.59%	10.71%	11.47%	−22.18%
Tobacco products	125.05	0.26%	145.77	0.36%	251.80	0.72%	174.21	0.42%	−28.22%	−16.32%	44.54%

Table 2 (continued)

35 Economic sectors of household expenditure (NACE classification)	Directors, seniors, qualified and intellectual workers (HL-1)		Technicians, clerks (HL-2)		Manual workers (HL-3)		Average, mean HE (adult eq.; 1.03)[a]		HL-1 (1.20)[a]	HL-2 (1.02)[a]	HL-3 (0.87)[a]
	€		€		€		€		Departure from the mean		
Trade and repair of motor vehicles	2,710.20	5.67%	2,639.28	6.50%	2,316.05	6.65%	2,555.17	6.22%	6.07%	3.29%	−9.36%
Transport by ship	56.49	0.12%	28.24	0.07%	39.96	0.11%	41.56	0.10%	35.92%	−32.06%	−3.85%
Transport equipment n.e.c.	104.67	0.22%	237.19	0.58%	119.22	0.34%	153.70	0.37%	−31.90%	54.33%	−22.43%
Water supply	158.28	0.33%	141.22	0.35%	167.60	0.48%	155.70	0.38%	1.66%	−9.30%	7.64%
Wearing apparel and furs	2,761.21	5.77%	2,137.56	5.26%	1,678.84	4.82%	2,192.54	5.34%	25.94%	−2.51%	−23.43%
TOTAL expenditures	47,823.45	100%	40,608.05	100%	34,844.12	100%	41,091.87	100%	16.38%	−1.18%	−15.20%

[a] Functional unit (€/h), allocation 22% of total annual time (=1,920 h)

for housing needs (e.g. water, electricity, heating, etc.), food, and transport. These relative contributions can fluctuate considerably between countries (Eurostat 2011). It therefore becomes important to evaluate the environmental impact difference due to each country-based HE variability.

3.3 Background life cycle inventory data and case studies

Since the foreground LCI data of HL's product system consist of HEs in the different economic sectors, the use of Environmentally Extended Input–Output Tables (EEIOTs) of EU-27, year 2005 (i.e. $EEIOT_{EU27}$), as implemented in SimaPro, is quite straightforward (Weidema et al. 2005). The use of a process based LCI, such as Ecoinvent, would be unfeasible due to the large amount of process-specific primary datasets that should be adapted to the different economic sectors. The IO database is the output of a Leontief-type environmentally extended, physical input–output model covering the EU-27. This model includes raw material extraction and processing of imported materials and waste treatment (Tukker et al. 2006). The benefits of using EEIO tables have been extensively discussed during the last decade (Suh and Huppes 2002; Suh and Huppes 2005; Mongelli et al. 2005; Suh and Nakamura 2007). $EEIOT_{EU27}$ expresses transactions in Euro, therefore it was modified to model production and export of the commodities used by "adult equivalents" that do work. $EEIOT_{EU27}$ is originally composed by a 60×60 IOT with aggregated extensions of direct and indirect raw materials uses (e.g. crude oil, gas, coal, sand and gravel, iron), emissions in air, soil and water, and land occupation. To model the import of commodities, we adopted the EEIOT of USA, year 2005 (i.e. $EEIOT_{USA}$), which is derived from a more detailed 480×480 IOT (Suh 2003).

The environmental extensions (EEs) included in the EEIOTs provide the necessary data per unit value (Euro or PPS) of economic sectors for the LCIA of HL. The EEs is directly related to a set of 35 goods and services as listed in Table 2. This set of commodities refers to the basket of household expenditures found in the statistical tables, which was manually built in SimaPro adopting the NACE classification code (see the Electronic Supplementary Material, Table S1).

4 Step II: LCIA of human labour

4.1 Environmental profiles of human labour types

Figure 3 compares the LCIA results calculated for the three types of HL in Luxembourg, based on the framework in Table 2. Midpoint characterization factors from ReCiPe method were used (see Table 1). The percentage values in Fig. 3 are related to the functional units of HL in euro/hour as described in Section 3.2 (see Electronic Supplementary Material—Table

Table 3 Average annual HE per adult equivalent (in PPS/year 2005) and correspondent PPP factors for the EU-27 countries; the functional units (in PPS/h) are calculated with a 22% allocation factor (refer to Section 3.1)

Country	Purchasing power standard (PPS/year)	Purchasing power parity (PPP)	Functional unit (PPS/h)
Austria	19344	1.06	0.48
Belgium	18831	1.09	0.47
Bulgaria	4213	0.74	0.11
Cyprus	17094	0.9	0.43
Czech Republic	6520	16.06	0.16
Denmark	16199	10.86	0.41
Estonia	6531	9.15	0.16
Finland	16082	1.23	0.40
France	17523	1.09	0.44
Germany	18952	1.03	0.47
Greece	17391	0.87	0.44
Hungary	6241	146.67	0.16
Ireland	20583	1.24	0.51
Italy	17663	1.07	0.44
Latvia	5316	0.36	0.13
Lithuania	5892	1.69	0.15
Luxembourg	32794	1.23	0.82
Malta	15108	0.69	0.38
Netherlands	19018	1.06	0.48
Poland	5817	2.2	0.15
Portugal	11674	0.84	0.29
Romania	2866	1.72	0.07
Slovakia	6517	0.64	0.16
Slovenia	13299	0.75	0.33
Spain	13940	0.91	0.35
Sweden	17414	11.48	0.44
United Kingdom	20047	0.76	0.50
EU-27	15225	1.00	0.38

S4 for absolute values). To illustrate the differences between HL types, we compared the results of the LCIA indicators, expressed in percentage, to the ratios between the reference functional units expressed in Table 2, i.e. 1.20 €/h in HL-1, 1.02 €/h in HL-2, and 0.87 €/h in HL-3 to which the 100%, 85%, and 73% were, respectively, assigned (see bars on "reference" in Fig. 3).

The impact of HL-3 increased relatively to the "reference" along with all LCIA indicators, i.e. from a minimum of 1% in the urban land occupation (ULO) to a maximum of 12% in

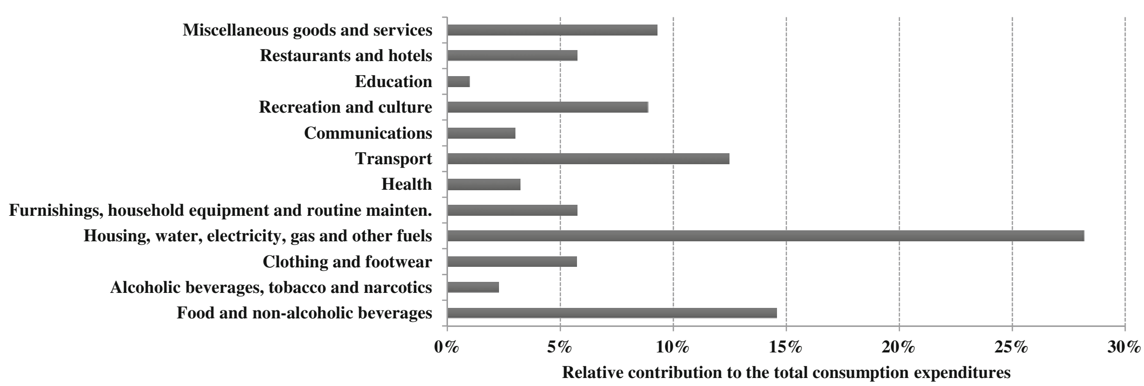

Fig. 2 Household expenditures distribution per adult equivalent in Europe-27 (year 2005, data source: Eurostat 2011); COICOP classification 1-digit for economic sectors is used

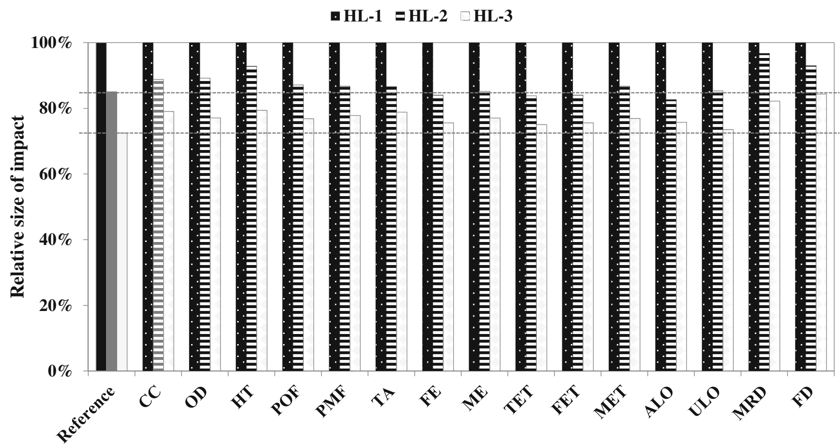

Fig. 3 Percentage comparison of the three human labour (HL) types in Luxembourg based on ReCiPe midpoint results and "reference" functional unit (in Euro/h)

fossil depletion (FD). In contrast, the impact of HL-2 substantially increased (between 8% and 12%) only for human toxicity (HT) and metal and fossil depletion potentials (MD and FD, respectively). It can be argued that workers of HL-1 spend (relatively) more money on household expenditures that generate (directly and indirectly) lower environmental impacts, such as "real estate services", whereas workers of HL-2 and, in particular, HL-3, spend less money overall and what they spend was on goods and services that provide higher impacts, e.g. "food products and beverages". However, on an absolute scale HL-1 always showed greater environmental impact than HL-2 and HL-3 (see Table S4 of the Electronic Supplementary Material). For example, 1 h of work in HL-1 may generate 0.52 kg CO_2-eq, whereas, under the same conditions, HL-2 and HL-3 then generate 0.46 and 0.41 kg CO_2-eq, respectively.

The relative contribution of HE sectors to (a) ULO and (b) FD is illustrated in Fig. 4. These two categories were taken as illustrative example to investigate the differences in HEs between the three types of HL. Indeed, FD and ULO were among the indicators with the highest and the lowest, respectively, variation from the "reference" (see Fig. 3).

The contribution of the economic sectors changed over the HLs types. For example, HL-3 showed higher relative contributions for the US sector of "agricultural, forestry and fishery services" in the ULO category than HL-1 and HL-2 (see Fig. 4a). ULO was more related to the HE for food than for other categories such as "other amusement and recreation services" or "freight forwarders and other transportation services" (see Fig. 4a). HL-3 has therefore a higher relative impact to ULO because of the higher consumption of food compared to HL-1 and HL-2. Furthermore, Fig. 4a highlights that the indirect contribution of imports (i.e. commodities from USA) to ULO was dominant for most of the goods and services consumed. HL-3 contained a higher relative contribution than HL-1 and HL-2 in the FD category for the European sectors "crude petroleum and natural gas" and "coal, lignite, peat". In general, the EEs for these sectors inventory more resources in the $EEIOT_{EU27}$ than in the $EEIOT_{USA}$ (see Fig. 4b). This can also explain why the relative impact of HL-3 increased when

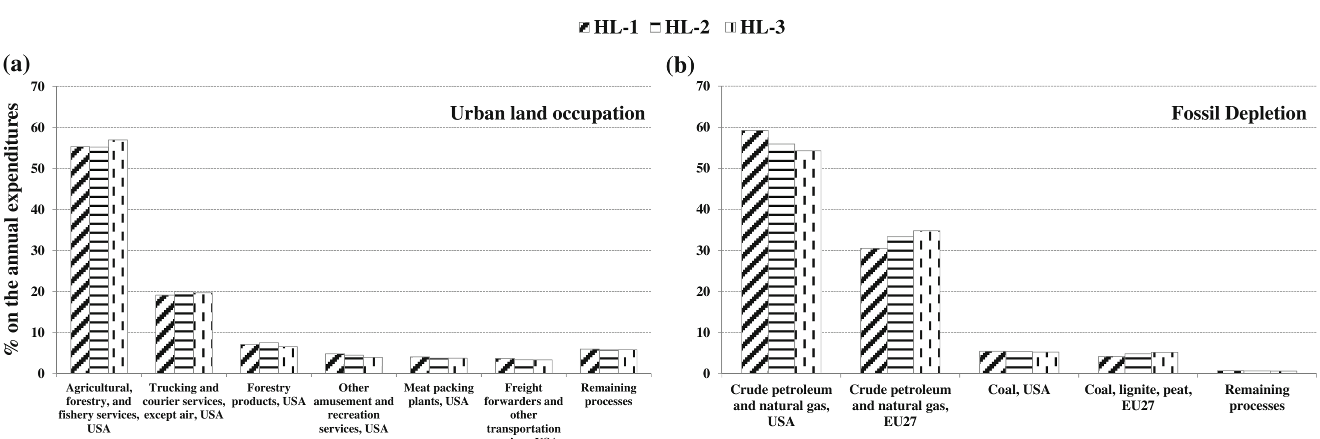

Fig. 4 Relative contribution of HE sectors to **a** urban land occupation (ULO) and **b** fossil depletion (FD) results, for the three HL types in Luxembourg (3% cut-off)

	CC	OD	HT	POF	PMF	TA	FE	ME	TET	FET	MET	ALO	ULO	MRD	FD	Mean contribution	SD (±)	Reference (PPS/hr)
Belgium	57%	50%	47%	55%	57%	60%	57%	59%	57%	57%	52%	61%	56%	42%	59%	55%	5%	57%
Bulgaria	25%	14%	14%	20%	24%	27%	24%	25%	24%	24%	20%	28%	21%	9%	24%	21%	5%	13%
Denmark	75%	50%	57%	67%	65%	69%	58%	64%	55%	58%	56%	57%	57%	46%	65%	60%	8%	49%
Estonia	30%	21%	20%	27%	30%	32%	29%	30%	29%	29%	25%	32%	27%	16%	27%	27%	5%	20%
Finland	55%	46%	48%	53%	54%	56%	50%	55%	50%	50%	49%	53%	54%	45%	47%	51%	3%	49%
France	64%	54%	53%	61%	63%	65%	59%	61%	60%	59%	56%	63%	60%	50%	62%	59%	4%	53%
Greece	59%	55%	48%	57%	66%	68%	75%	70%	75%	76%	66%	83%	68%	42%	54%	64%	11%	53%
Ireland	71%	65%	60%	67%	74%	76%	78%	78%	78%	78%	71%	84%	74%	56%	70%	72%	7%	63%
Italy	62%	52%	46%	59%	66%	71%	68%	71%	69%	68%	60%	75%	63%	40%	64%	62%	10%	54%
Lithuania	33%	23%	20%	29%	35%	38%	36%	37%	36%	36%	30%	42%	32%	14%	29%	31%	8%	18%
Luxembourg	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	0%	100%
Malta	62%	64%	57%	61%	69%	71%	74%	72%	76%	74%	69%	83%	73%	56%	46%	67%	10%	46%
Slovenia	54%	43%	43%	49%	53%	55%	53%	54%	53%	53%	49%	56%	50%	39%	60%	51%	6%	41%
Spain	45%	39%	35%	43%	51%	54%	57%	55%	58%	57%	49%	64%	52%	31%	38%	49%	10%	43%
United Kingdom	58%	53%	52%	58%	60%	60%	58%	60%	60%	58%	56%	63%	61%	50%	54%	58%	4%	61%

Fig. 5 LCIA results (ReCiPe midpoint, Europe H, see Table 1 for acronyms) as compared to the reference functional unit (in PPS/h) of each average human labour within a sample of 15 out of the EU-27 countries analysed; the relative contributions relate to the highest values recorded for the Luxembourg case, to which the 100% is assigned

compared to the ratios of reference among functional units as aforementioned (see Fig. 3).

4.2 Human labour in European countries

Figure 5 shows the LCIA results of the average HL for 15 out of the EU-27 countries analysed. These were selected among the most complete in terms of HEs as evaluated by Eurostat for the year 2005 (Eurostat 2011; see also the Electronic Supplementary Material, Table S2). The relative contributions are shown instead of absolute values in order to illustrate the spatial and behavioural differences of HL among the 15 countries. Results are expressed in percent with respect to Luxembourg, which always showed the highest values and are compared to the correspondent reference functional units (in PPS/h, see Table 3).

The greatest values for the HL in Luxembourg were mainly due to an overall relatively different behaviour in consumption expenditures to those of the other countries. More specifically, the “adult equivalent” in Luxembourg gave more weight than other countries to the sectors of “real estate services”, “hotel and restaurant”, and “manufacture of motor vehicles, trailers” rather than to the sector of “food product and beverages” (see Electronic Supplementary Material, Table S5). Furthermore, the money spent (in PPS) per hour of work in Luxembourg were greater than in other countries and were often more than double (see functional unit in Table 3).

The standard deviation of the percentage scores showed a greater variability for Greece, Italy, Malta, and Spain (SD≥ 10%). For these countries, a significant increase of the relative impact for the category ALO was observed, which depends on the indirect production and import of food products. When examining the countries with the lowest functional unit (see column “reference”), e.g. Bulgaria and Lithuania, a large increase in the relative impact of many categories was evident. However, the HL in Malta showed the highest deviation from the functional unit, in particular with regard to the ecotoxicity (i.e. TET, FET, MET) and eutrophication (i.e. FE, ME) potentials, which mainly originate from the life cycle of agricultural products in Europe.

The contribution of the HE sectors to the different HL systems, per country, was analysed in detail. Figure 6 shows 4 out of the 15 LCIA indicators calculated (CC, FD, HT, and ALO), while the complete set of results is reported in the Electronic Supplementary Material (Fig. S1). The results in Fig. 6 show that significant differences did exist among the 15 countries. For example, while in Denmark (DK) and Estonia (EE) the contribution of HL to climate change (CC) was mainly due to the consumption of “electricity, gas, steam and hot water, EU27”, in the other countries it was due to many minor contributions of goods and services included in the category “remaining processes” (i.e. set of sectors with individual contribution ≤2%). The highest contribution of the category “remaining processes” in all countries (>29%) was observed for the LCIA category HT. Opposite results were observed for the other two LCIA categories represented in Fig. 6. Only a few of HEs contributed to the overall impact: the consumption of “crude oil and natural gas” (from US import and produced within Europe) mostly caused fossil

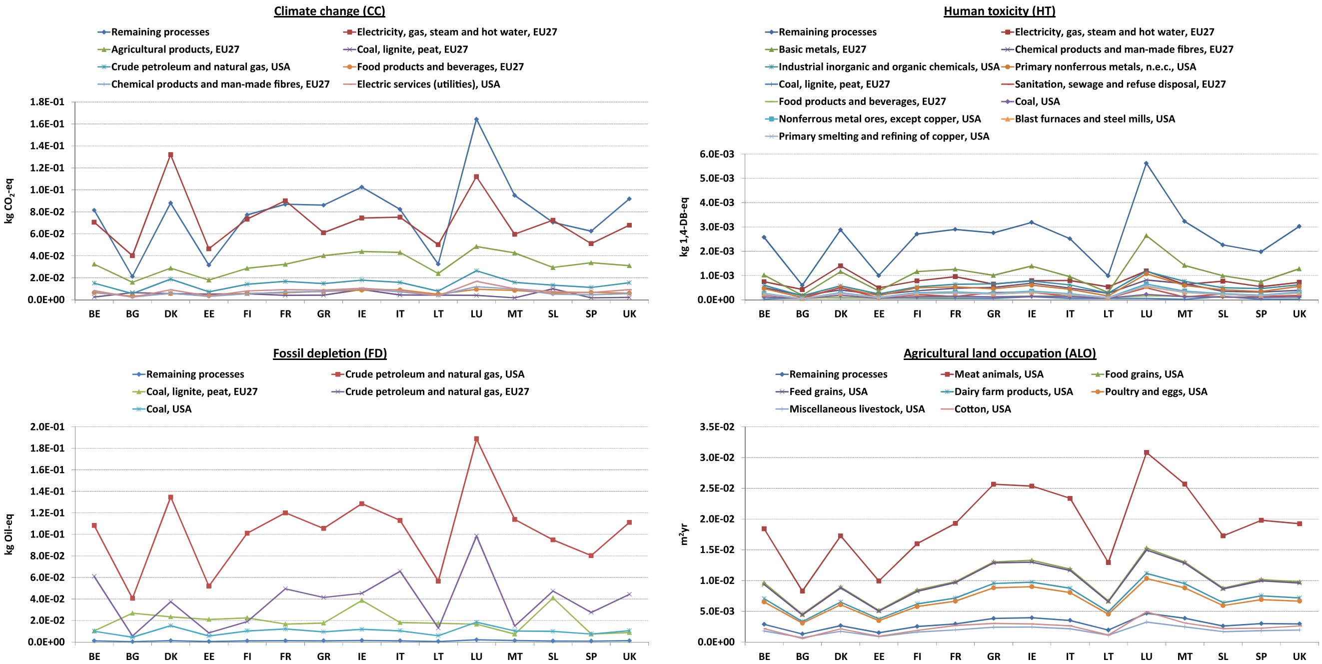

Fig. 6 Contribution of the household expenditures to the average HL in 15/EU-27 countries in four LCIA indicators (ReCiPe midpoint, Europe H); data refer to PPS/h per adult equivalent; a cut-off of 2% has been applied; refer to Table S3 of the Electronic supplementary material for the country names and codes (ISO 3166-1)

depletion, and the production of "meat animals", "food grains", and "feed grains" (in USA), contributed to the impact on agricultural land occupation.

These different contribution patterns could be explained by the fact that EEs include more emission than resource elementary flows. Consequently, some goods and services of (apparently) minor importance for household needs may have assigned a greater impact for emission-related LCIA categories (e.g. CC and HT) rather than for resource-related categories (e.g. FD and ALO) since they include a more comprehensive set of direct and indirect process emissions (see also Fig. S1 in the Electronic Supplementary Material).

5 Step III: application of the HL accounting framework to case studies

5.1 Case studies

Ten case studies were selected from the literature on eMergy (Brandt-Williams 2002; Paoli et al. 2008; Pulselli et al. 2008; Vassallo et al. 2009; Pulselli et al. 2011; Ciotola et al. 2011) according to the quality and completeness of the inventory data. Emergy is an environmental accounting method that has historically tackled the assessment of the contribution of HL in the overall assessment of products and processes. The inventory phase in an eMergy evaluation is similar to an LCI at the foreground level (Ulgiati et al. 2006; Rugani et al. 2011). Inventory data on HL taken from five cases of industrial production (i.e. cement, tap water, wastewater treatment, photovoltaic module, and biogas) and five cases of agri-food production (i.e. potatoes, sugarcane, soybeans, cotton, and corn-grain cultivations) were added to corresponding unit processes collected from the Ecoinvent database v2.2 (Ecoinvent 2010), as illustrated in Table 4, to obtain consistent LCI models. The contribution of HL was then quantified by applying the methodological framework illustrated in Section 3 (IOT-based database) coupled to the rest of the inventory (Ecoinvent-based) to obtain a hybrid LCI (Crawford 2008; Fig. 7).

HL input data collected in working hours were first converted to Euro. The conversion from hours to Euro was performed using the factor 1.03 €/h, which relates to the functional unit of the "Luxembourgish (adult-eq.)" human labour (see Table 2). The Luxembourg's average EE-IO profile was used because of the lack of information provided by the eMergy studies, where only total working hours were available without any specification about the work skills. The Ecoinvent datasets used for non-HL-related inventory data are representative of the Swiss/European (for the industrial processes) and USA/Brazil (for the agri-food processes) conditions (see Table 4), instead data for HL refers to mixed conditions (e.g. USA, Italy) which are often not clearly reported. For a preliminary assessment of HL, it is assumed that this geographical inconsistency does not significantly affect the comparison between HL and non-HL contributions to LCIA results. Finally, it must be pointed out that the functional unit of human labour in Luxembourg is the highest (between 38% to 91% in terms of PPS/h) among the

Table 4 Overview on the implementation of human labour in the case studies selected from the eMergy literature

Case studies	Abbreviation	Emergy source for HL data (in h or J)	Reference production	Unit	HL (total h)/reference production	Modified Ecoinvent process name	Ecoinvent FU	€/FU[a]	Further information used to calculate hours of HL
Industrial sector									
Cement production	CEM	Pulselli et al. (2008)	7.15E+08	kg/year	1.04E+05	Cement, unspecified, at plant/CH U	1 kg	1.50E-04	Assumed (1,740 h×60 workers×125 kcal/h×4.186 J/kcal)=5.42E10 J/year
Tap water distribution	TAP	Pulselli et al. (2011)	1.16E+07	m^3/year	8.76E+03	Tap water, at user/RER U	1 kg	7.79E-07	Average values calculated among three plants
Wastewater treatment process	WWT	Vassallo et al. (2009)	1.19E+07	m^3/year	2.19E+05	Treatment, sewage, to wastewater treatment, class 1/CH U	1 m^3	1.90E-02	Assumed metabolism energy: 4.36E+05 J/h (from 2,500 kcal/day×4,186 J/kcal)
Photovoltaic module production	PVM	Paoli et al. (2009)	6.35E-01	m^2/panel	3.12E+00	Photovoltaic panel, single-Si, at plant/RER/I U	1 m^2	5.07E+00	"
Biogas production	BIO	Ciotola et al. (2011)	7.92E+03	m^3/year	1.59E+02	Biogas, from slurry, at agricultural co-fermentation, covered/CH U	1 m^3	2.07E-02	"
Agri-food sector									
Potatoes production	POT	Brandt-Williams (2002)	5.43E+03	kg/ha/year	3.14E+02	Potatoes, at farm/US U	1 kg	5.96E-02	"
Sugarcane production	SUG	Brandt-Williams (2002)	2.27E+04	kg/ha/year	3.14E+01	Sugarcane, at farm/BR U	1 kg	1.43E-03	"
Soybeans production	SOY	Brandt-Williams (2002)	4.04E+02	kg/ha/year	1.68E+01	Soybeans, at farm/US U	1 kg	4.29E-02	"
Cotton production	COT	Brandt-Williams (2002)	7.38E+02	kg/ha/year	2.04E+02	Cotton fibres, at farm/US U	1 kg	2.85E-01	"
Corn, grain production	COR	Brandt-Williams (2002)	9.17E+02	kg/ha/year	3.03E+01	Corn, at farm/US U	1 kg	3.40E-02	"

[a] Value implemented within the Ecoinvent processes; derived using the conversion factor of 1.03 €/h (average Luxembourgish human labour; see Table 2)

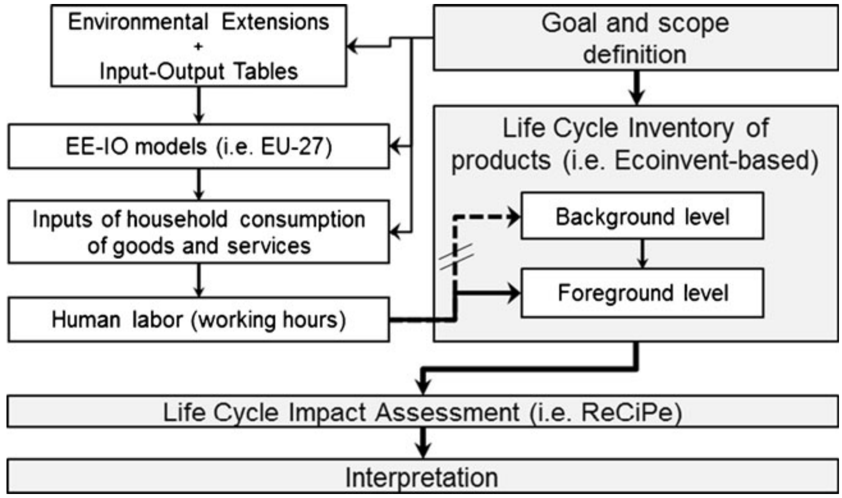

Fig. 7 Framework developed in this study for hybrid LCA including human labour

European countries (see the Electronic Supplementary Material, Table S2). Therefore, it is likely that the use of this EE-IO profile will provide the highest possible results.

5.2 LCIA results from the hybrid analysis

Figure 8 compares the relative average HL contribution to the total environmental impact (per impact category) of the two set of LCA case studies (i.e. average scores of, respectively, 5 industrial and 5 agri-food processes as illustrated in Table 4). The assessment results show that human labour significantly contributed to the total impact on several categories, both in industrial and agri-food production sectors. For example, quite high contributions (between 9% and 20%) were observed for OD, FD, ALO, and TET (see Fig. 8).

Results also show that, on average, the HL contribution within the agri-food sector was larger than within the industrial sector for most of the impact categories, ranging between 0.4% (i.e. ME) to 10.5% (i.e. OD). Exceptions were found for three impact categories, i.e. FET, ALO, and TET, where the HL contribution within the industrial sector was higher.

The relative contribution calculated for each individual case study is reported in Table S6 of the Electronic Supplementary Material. Concerning the industrial sector, the cement production and the tap water distribution processes showed the lowest contributions of the human labour to the overall set of impact categories (<1.1%), while the highest contributions were found for biogas production (with 49% on TET and 43% on FD). Some interesting values are also observed for the case of wastewater treatment, where HL contributes by more than 10% to OD and FD, and up to 35% to the ALO.

Conversely to the industrial processes, results for the agri-food sector were rather homogeneous, mostly because of the data source used, which considered similar agricultural practices and inventory data (see Table 4). For example, all the impact categories of ecotoxicity, eutrophication, and agricultural land occupation were not significantly affected by HL (contributions <1%, with very few exceptions,

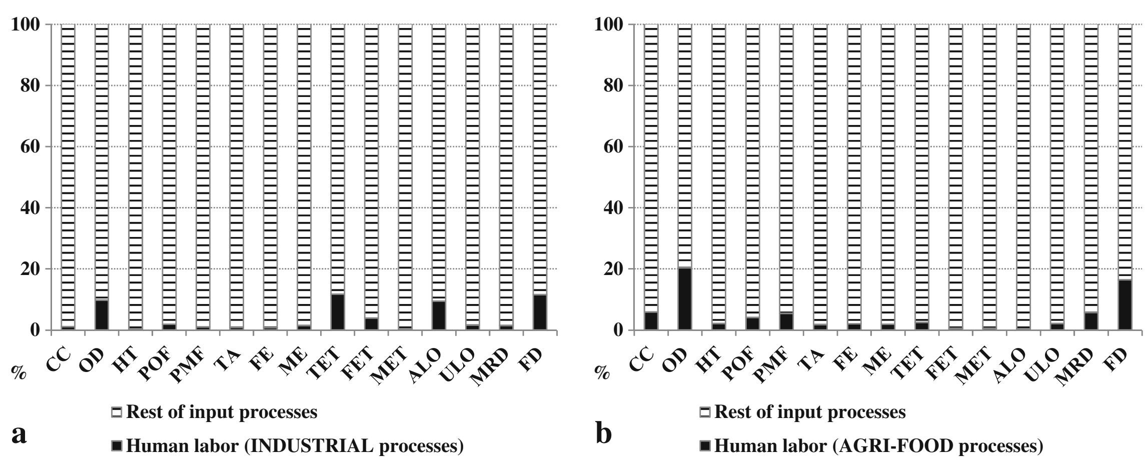

Fig. 8 Relative average contribution of human labour to the total environmental impact calculated (ReCiPe midpoint categories, see Table 1); the average contribution is calculated from the relative contributions of the "Luxembourgish" human labour (average profile, see Table 2) implemented for **a** 5 industrial case studies and **b** 5 agri-food case studies, as illustrated in Table 4

see Table S6 in the Electronic Supplementary Material), whereas those impacts were likely to depend more on other inputs such as the use of fertilizers. Other categories like FD, MD, CC, PMF, and OD, which may be more related to the HL system boundaries (e.g. food production and supply, use of fuels, etc.), showed much higher values of HL contributions (between 2% and 25% among all the five case studies). The highest values were found for soybeans production (up to 34% to the OD and FD).

6 Discussion

This paper provides a methodological framework to account for human labour in LCA and an application in several case studies. Different human labour types were set by including direct inputs of purchased goods and services necessary for human activity purposes, such as food and drinks, water for non-drinking uses, means of transport, clothes, household and energy consumptions, and so forth. Inventory models based on an environmentally extended input–output database (EU-27) were built and coupled to process based LCIs of industrial and agri-food products. Results showed that the contribution of human labour in the LCA of these products must not be neglected, since the results obtained are relevant for some environmental impact categories such as fossil and ozone depletions.

While the outcomes of the study put new light on the issue of environmental impact due to humans working activity, both methodological and conceptual aspects of the present research need to be further discussed. Despite the fact that the results do confirm the added value obtained by including human labour in LCA, several limitations and constraints exist, mostly due to data and models used.

The quality of the results depends on the number and type of working hours considered. This data comes from past eMergy evaluation studies (see Table 4). Those studies were selected because of their remarkable completeness in the inventory of process inputs, which is normally a critical aspect for eMergy (Hau and Bakshi 2004). However, in eMergy a differentiation of HL in terms of knowledge (e.g. different classes by education; Odum 1988) or work typology (e.g. manual and intellectual workers; Ortega 1997) is usually provided. Despite these features, which make eMergy one of the most straightforward and preeminent methods to evaluate HL, the selected sources included only an “undefined” type of human labour within their eMergy evaluation tables (see Section 5.1). This is certainly a restriction to the specificity and representativeness of the case studies, which should be overcome at the level of LCI process data collection.

A major limitation dealt with a truncation error associated with the completeness of the system boundary, since the inputs of HL were only implemented at the foreground level in the case studies. Results obtained using an LCI database (or an EEIO model) including labour at the background level (each product or industry sector, as illustrated by the broken flow in Fig. 7) would increase the impact of HL even further beyond the magnitude established by this study, which would be evidence for this kind of truncation error. In this connection, we have attempted to account for HL also at higher levels of the economy, i.e. upstream in the IO sectors of the EU-27 framework used in this study. We accounted for the human labour input in the production of the commodities directly and indirectly consumed by the human labour input required to produce the ten products (see Table 4). As a result, the total environmental impacts (see Fig. 8 and Table S6 in the Electronic Supplementary Material) is increased by maximum 2% (see Tables S7 and S8 in the Electronic Supplementary Material for further details), which is inherently a negligible contribution. This approach is conceptually similar to other approaches adopted in literature (e.g. Lenzen and Treloar, 2003), in so far enabling to conclude that not only first-order but also higher-order requirements of human labour shall be incorporated in the LCI boundaries to properly evaluate the feedback and multiplier contribution for different stressors that may lie within different production layers. Therefore, for the completeness of our hybrid framework and for its future application and improvement, including HL inputs within the entire database of LCI processes (e.g. Ecoinvent) would be recommended, along with process-specific differentiations of labour.

The consistency of the EEIO models is another issue to investigate for further developments of the methodology. As already pointed out (Heijungs and Suh 2006; Rowley et al. 2009), the use of IO models may lead to several assumptions and limitations, including: uncertainties in the aggregation of goods and services due to the variability of industry classification schemes; generalization in the assignment of environmental burdens (i.e. impacts arising abroad from the production of imports are identical to those generated by equivalent domestic production); misallocation of environmental impacts when, under the industry-technology assumption, each industry that produces more than one product is generally assumed to produce only one homogeneous product (i.e. all products within a certain sector finally show the same environmental impact per unit of currency); and uncertainties due to the large variability of production technologies in the national datasets adopted. Other criticisms can be attributed to the EE inventories, which were fairly incomplete for this analysis. Indeed, only iron, aluminum, sand, and gravel, two generic kinds of land (i.e. industrial and agricultural), and fossil fuels such as oil, coal, and gas were considered, while several other minerals and metals used, e.g. for infrastructures, or additional occupied/transformed land types, all water consumptions, and all inputs of biomass and renewable energy carrier exploited, e.g. for electricity and heat generation, are not included within the list of raw materials of the EEs

adopted. Furthermore, wastes and end-of-life processes have not been assessed in our IO models, leaving out the consideration of the related meaningful environmental burdens (e.g. wastewater and body excretions treatments, infrastructure dismantling, etc.; Muñoz et al. 2008; Muñoz et al. 2010). The exclusion of all these resources and processes from the inventory may have led to the underestimation of HL contribution to the environmental impact. The relevance of waste treatment phases in LCA and IO is undoubtedly high (Lin 2009; Nakamura and Kondo 2002; Kondo and Nakamura 2004) and those processes should definitively be addressed in a further refinement of the present calculation approach. Accordingly, new detailed EEIOTs that will be available in the near future (EXIOPOL 2011) should be integrated within the present framework.

Finally, we do recognize that the representativeness of the HL types can be questionable as well. Indeed, only three main labour types were set based on country average lifestyles (i.e. household expenditures of Luxembourg). In our point of view, however, this was the best way to provide an extensive characterization of the human labour while enabling comparison among different EU countries through PPS values.

7 Conclusions and outlook

The outcomes of this work can provide significant improvements to the LCA methodology from both a conceptual and a technical point of view. Indeed, quantifying the contribution of HL like a conventional flow, product, or resource makes the perception of humans as leading actors of environmental impact, with explicit responsibility for their actions. Furthermore, though for many technological production chains the physical HL contribution could be negligible (at least when compared with that of machineries; Pimentel 1993; Zhang and Dornfeld 2007), there might be cases where human work essential and largely contributes to the life cycle impact (e.g. handmade manufacturing processes or most of agricultural productions). Apart from the physical entity of this contribution, what is important to note is the inherent conceptual mistake that occurs if neglecting (as is done now) an input (i.e. the human labour) from a process that would not exist in the reality without that input: an eco-profile of human labour should always be added to life cycle models that entail a direct human contribution.

Despite the limitations detailed in Section 6, results from this study suggest several lines of reasoning on the importance of HL for LCA, which could influence future developments:

1) The addition of human labour in the LCI of a product or process can improve the accuracy of the whole analysis, since its contribution to the overall LCIA could be significant. So far, the methodological framework here presented can be used for a future implementation of human labour at a level of unit process in LCA (e.g. within the Ecoinvent database). The inclusion of the "background" requirements of HL in the EEIO models (thus accounting for higher-layers' effects) is recommended to increase the consistency of the entire system boundary. Indeed, only the direct and indirect effects resulting from initial changes in the economic output are usually considered, which have impact at the foreground and in the backward-linked sectors, respectively. However, several studies recognize the relevance of a third or "induced" effect (included in the so called type-II multiplier in IO), which may occur when households spend some of their additional income (derived by the direct and indirect effect) on goods and services increasing the multiplier effect (e.g. Miller 1980; Lenzen and Schaeffer 2004). An operational use of this multiplier IO technique for hybrid LCA including labour was beyond the scope of this paper but should be further explored to describe the "hidden" human labour interconnections among infinite supply-chains.
2) By using a hybrid approach, possible double counting with non-human (i.e. machine driven) labour is easily avoided, since all processes required to produce and operate the machineries are already included in the process-LCA, while the EEIO model with household expenditures covers only the direct inputs from human needs; we advocate that the use of HEs instead of salary or income quantities remains the recommended implementation and an added value to account for the impact of human labour using EEIO models. Though data on income can be extensively available in IO models, accounting for human consumption behaviours through the earnings as a total appears to produce incorrect results, since the wage could not be completely spent for the worker's consumptions.
3) The evaluation of HL using an LCA approach allows identifying opportunities to reduce the environmental impact along with the most critical processes of the human life cycle, such as alternative strategies in food supply or in the usage of transport means.
4) Considering HL as a part of a product system helps to reduce the distance between biosphere and technosphere, since humans are put at the same level of the technological/economic activities. In addition, the evaluation of HL within the technosphere leads to a significant conceptual shift from a mere utilitarian and anthropocentric perspective to a more ecology-oriented one, where humans not only pay for the ecosystem services/resources they use and emissions/wastes they release, but are direct cause for, and components of the affected environment.

5) The human labour is intrinsically linked to the economic and social aspects of a life cycle. So far, additional cost and social/organizational data of HL might be integrated in LCIs to provide useful information and an added value for more comprehensive assessment of the real life cycle cost (e.g. through addition of salaries and wages to the Life Cycle Costing analysis) or social quality factors of labour (e.g. through implementation of further labour impact categories in the SLCA) associated to a production chain.
6) Here, only physical flows of material and energy requirements were included in the evaluation of HL. However, humans are not machineries and they are driven by flows of information, knowledge, educational and cultural experience, and so forth. These are essential items for our future sustainable development that should definitively be integrated into human labour LCI profiles, which remains an open task.

Acknowledgements This work is supported by the National Research Fund of Luxembourg and cofounded under the Marie Curie Actions of the European Commission (FP7-COFUND). These are gratefully acknowledged. We thank Dr. Alex Cornelissen for the English proof-reading and Colin Jury for his helpful comments on the preliminary idea behind this paper. We would like also to thank three anonymous reviewers for their helpful comments, which notably contributed to improve the quality of this paper.

References

Abel T (2010) Human transformities in a global hierarchy: emergy and scale in the production of people and culture. Ecol Model 221 (17):2112–2117

Bender MH (2003) Energy budget of an energy-integrated organic farm. In: Proceedings of organic agriculture research symposium 2003, ASA-CSSA-SSSA Annual Meetings, Denver, Colorado, USA

Brandt-Williams S (2002) Handbook of emergy evaluation—folio#4—Emergy of Florida Agriculture. Centre for Environmental Policy, University of Florida, Gainesville, USA. http://www.emergysystems.org/downloads/Folios/Folio_4.pdf. Accessed 19 September 2011

Brown MT, Herendeen RA (1996) Embodied energy analysis and emergy analysis: a comparative view. Ecol Econ 19(3):219–235

Campbell DE (1998) Emergy analysis of human carrying capacity and regional sustainability: an example using the State of Maine. Environ Monit Assess 51(1–2):31–569

Ciotola RJ, Lansing S, Martin JF (2011) Emergy analysis of biogas production and electricity generation from small-scale agricultural digesters. Ecol Eng 37(11):1681–1691

Claro RM, Levy RB, Bandoni DH, Mondini L (2010) Per capita versus adult-equivalent estimates of calorie availability in household budget surveys. Cad Saude Publica 26(11):2188–2195

Crawford RH (2008) Validation of a hybrid life-cycle inventory analysis method. J Environ Manage 88(3):496–506

Da Costa R, de Laiglesia JR, Martínez E, Melguizo A (2011) The economy of the possible pensions and informality in Latin America. Working Paper No. 295, OECD Development Centre, Paris, France. http://www.oecd.org/dataoecd/45/22/46937116.pdf. Accessed 19 September 2011

Dreyer LC, Hauschild MZ, Schierbeck J (2010) Characterisation of social impacts in LCA part 1: development of indicators for labour rights. Int J Life Cycle Assess 15(3):247–259

Druckman A, Jackson T (2009) The carbon footprint of UK households 1990–2004: a socio-economically disaggregated, quasi-multiregional input–output model. Ecol Econ 68(7):2066–2077

Druckman P, Sinclair P, Jackson T (2008) A geographically and socio-economically disaggregated local household consumption model for the UK. J Clean Prod 16(7):870–880

Ecoinvent (2010) Ecoinvent database v2.2. Swiss Centre for Life-Cycle Inventories, Dübendorf, Switzerland. http://www.ecoinvent.org/database/. Accessed 19 September 2011

Eurostat (2008a). European price statistics—an overview. Eurostat statistical books, European Commission, Luxembourg. http://epp.eurostat.ec.europa.eu/cache/ITY_OFFPUB/KS-70-07-038/EN/KS-70-07-038-EN.PDF. Accessed 19 September 2011

Eurostat (2008b) NACE Rev.2—Statistical classification of economic activities in the European Community. Methologies and Working papers, Eurostat-European Commission, Luxembourg. http://circa.europa.eu/irc/dsis/nacecpacon/info/data/en/NACE%20Rev%202%20structure%20+%20explanatory%20notes%20-%20EN.pdf. Accessed 19 September 2011

Eurostat (2011) Mean consumption expenditure by detailed COICOP level (in PPS), year 2005. http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/search_database. Accessed 19 September 2011

Eurostat-OECD (2006) Methodological manual on purchasing power parities. Office for Official Publications of the European Communities, Luxembourg. http://www.oecd.org/dataoecd/59/10/37984252.pdf. Accessed 19 September 2011

EXIOPOL (2011) A new environmental accounting framework using externality data and input–output tools for policy analysis. http://www.feem-project.net/exiopol/index.php. Accessed 19 September 2011

Fluck RC (1981) Net energy sequestered in agricultural labor. T Am Soc Agr Eng 24(6):1449–1455

Fluck RC (1992) Energy of human labor. In: Fluck RC (ed) Energy in Farm production, vol. 6. of energy in world agriculture. Elsevier, Amsterdam, pp 31–36

Giampietro M (1997) Socioeconomic pressure, demographic pressure, environmental loading and technological changes in agriculture. Agr Ecosys Environ 65(3):201–229

Giampietro M, Pimentel D (1990) Assessment of the energetics of human labour. Agr Ecosys Environ 32(3–4):257–272

Giampietro M, Mayumi K, Munda G (2006) Integrated assessment and energy analysis: quality assurance in multi-criteria analysis of sustainability. Energy 31(1):59–86

Goedkoop M, Heijungs R, Huijbregts MAJ, de Schryver A, Struijs J, van Zelm R (2009) ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition, report I: characterisation. The Hague: Ministry of VROM, The Netherlands

Guzmán GI, Alonso AM (2008) A comparison of energy use in conventional and organic olive oil production in Spain. Agr Syst 98(3):167–176

Haberl H (2001a) The energetic metabolism of societies—part I: accounting concepts. J Ind Ecol 5(1):11–33

Haberl H (2001b) The energetic metabolism of societies—part II: empirical examples. J Ind Ecol 5(2):71–88

Haberl H, Weisz H, Amann C, Bondeau A, Eisenmenger N, Erb K-H, Fischer-Kowalski M, Fridolin Krausmann F (2006) The energetic metabolism of the European Union and the United States decadal energy input time-series with an emphasis on biomass. J Ind Ecol 10(4):151–171

Hau JL, Bakshi BR (2004) Promise and problems of emergy analysis. Ecol Model 178(1–2):215–225

Heijungs R, Suh S (2006) Reformulation of matrix-based LCI: from product balance to process balance. J Cleaner Prod 14(1):47–51

Hertwich EG (2005) Life cycle approaches to sustainable consumption: a critical review. Environ Sci Technol 39(13):4673–4684

Hertwich EG (2011) The life cycle environmental impacts of consumption. Econ Sys Res 23(1):27–47

Ingwersen WW (2010) Uncertainty characterization for emergy values. Ecol Model 221(3):445–452

Jørgensen A, Lai LCH, Hauschild MZ (2010) Assessing the validity of impact pathways for child labour and well-being in social life cycle assessment. Int J Life Cycle Assess 15(1):5–16

Kondo Y, Nakamura S (2004) Evaluating alternative life-cycle strategies for electrical appliances by the waste input–output model. Int J Life Cycle Assess 9(4):236–246

Krausmann F (2004) Milk, manure, and muscle power. Livestock and the transformation of preindustrial agriculture in Central Europe. Hum Ecol 32(6):735–772

Krausmann F, Haberl H (2002) The process of industrialization from the perspective of energetic metabolism—socioeconomic energy flows in Austria 1830–1995. Ecol Econ 41(2):177–201

Lenzen M, Peters G (2010) How city dwellers affect their resource hinterland—a spatial impact study of Australian households. J Ind Ecol 14(1):73–90

Lenzen M, Schaeffer R (2004) Environmental and social accounting for Brazil. Environ Res Econ 27(2):201–226

Lenzen M, Treloar G (2003) Differential convergence of life-cycle inventories toward upstream production layers implications for life-cycle assessment. J Ind Ecol 6(3–4):137–160

Lin C (2009) Hybrid input–output analysis of wastewater treatment and environmental impacts: a case study for the Tokyo Metropolis. Ecol Econ 68(7):2096–2105

Loake C (2001) Energy accounting and well-being—examining UK organic and conventional farming systems through a human energy perspective. Agr Syst 70(1):275–294

Milà L, Domènech X, Rieradevall J, Fullana P, Puig R (1998) Application of life cycle assessment to footwear. Int J Life Cycle Assess 3(4):203–208

Miller RE (1980) The relationship between type I and type II income multipliers in an input–output model: a comment. Int Regional Sci Rev 5(2):185–188

Moll HC, Noorman KJ, Kok R, Engström R, Throne-Holst H, Clark C (2005) Pursuing more sustainable consumption by analyzing household metabolism in European countries and cities. J Ind Ecol 9(1):259–276

Mongelli I, Suh S, Huppes G (2005) A structure comparison of two approaches to LCA inventory data, based on the MIET and ETH databases. Int J Life Cycle Assess 10(5):317–324

Munksgaard J, Wier M, Lenzen M, Dey C (2005) Using input–output analysis to measure the environmental pressure of consumption at different spatial levels. J Ind Ecol 9(1–2):169–185

Muñoz I, Milà i Canals L, Clift R (2008) Consider a spherical man—a simple model to include human excretion in life cycle assessment of food products. J Ind Ecol 12(4):521–538

Muñoz I, Milà i Canals L, Fernández-Alba AR (2010) Life cycle assessment of the average Spanish diet including human excretion. Int J Life Cycle Assess 15(8):794–805

Nakamura S, Kondo Y (2002) Input–output analysis of waste management. J Ind Ecol 6(1):39–63

Nebel B, Zimmer B, Wegener G (2006) Life cycle assessment of wood floor coverings—a representative study for the German flooring industry. Int J Life Cycle Assess 11(3):172–182

Nguyen TLT, Gheewala SH (2008a) Life cycle assessment of fuel ethanol from cane molasses in Thailand. Int J Life Cycle Assess 13(4):301–311

Nguyen TLT, Gheewala SH (2008b) Life cycle assessment of fuel ethanol from cassava in Thailand. Int J Life Cycle Assess 13 (2):147–154

Nguyen TLT, Gheewala SH, Garivait S (2007) Full chain energy analysis of fuel ethanol from cassava in Thailand. Environ Sci Technol 41 (11):4135–4142

Nijdam DS, Wilting HC, Goedkoop MJ, Madsen J (2005) Environmental load from Dutch private consumption. J Ind Ecol 9(1–2): 147–168

Odum HT (1988) Self-organization, transformity, and information. Science 242(4882):1132–1139

Odum HT (1996) Environmental accounting: emergy and environmental decision making. Wiley, New York, USA

Ortega E (1997) Handbook of emergy calculation. http://www.unicamp.br/fea/ortega/curso/handbook.htm. Accessed 19 September 2011

Paoli C, Vassallo P, Fabiano M (2008) Solar power: an approach to transformity evaluation. Ecol Eng 34(3):191–206

Pimentel D (1993) Economics and energetics of organic and conventional farming. J Agric Environ Ethics 6(1):53–60

Pulselli RM (2010) Integrating emergy evaluation and geographic information systems for monitoring resource use in the Abruzzo region (Italy). J Environ Manage 91(11):2349–2357

Pulselli RM, Simoncini E, Ridolfi R, Bastianoni S (2008) Specific emergy of cement and concrete: an energy-based appraisal of building materials and their transport. Ecol Indic 8(5):647–656

Pulselli FM, Patrizi N, Focardi S (2011) Calculation of the unit emergy value of water in an Italian watershed. Ecol Model 222(16):2929–2938

Rowley HV, Lundie S, Peters GM (2009) A hybrid life cycle assessment model for comparison with conventional methodologies in Australia. Int J Life Cycle Assess 14(6):508–516

Rugani B, Huijbregts MAJ, Mutel C, Bastianoni S, Hellweg S (2011) Solar energy demand (SED) of commodity life cycles. Environ Sci Technol 45(12):5426–5433

Sciubba E, Ulgiati S (2005) Emergy and exergy analyses: complementary methods or irreducible ideological options? Energy 30(10):1953–1988

Sciubba E, Bastianoni S, Tiezzi E (2008) Exergy and extended exergy accounting of very large complex systems with an application to the province of Siena, Italy. J Environ Manage 86(2):372–382

Silalertruksa T, Gheewala SH (2009) Environmental sustainability assessment of bio-ethanol production in Thailand. Energy 34 (11):1933–1946

Sorrells NR, Pimentel D (1981) Food, energy, and the environment: alternatives for creating new lifestyles. Am Biol Teach 43(4):190–195

Spirinckx C, Ceuterick D (1996) Biodiesel and fossil diesel fuel: comparative life cycle assessment. Int J Life Cycle Assess 1 (3):127–132

STATEC (2008) Dépenses annuelles moyennes par ménage selon la catégorie socio-professionnelle de la personne de reference. http://www.statistiques.public.lu/stat/TableViewer/tableViewHTML.aspx?ReportId=517&IF_Language=fra&MainTheme=3&FldrName=1&RFPath=28. Accessed 19 September 2011

UN Statistics Division (2011) COICOP—classification of individual consumption according to purpose. http://unstats.un.org/unsd/cr/registry/regcst.asp?Cl=5. Accessed 19 September 2011

Suh S (2003) MIET 3.0 User Guide—an inventory estimation tool for missing flows using input–output techniques. SimaPro Database Manual, PRé Consultants, The Netherlands. http://www.pre.nl/download/manuals/DatabaseManualUSAIODatabase98.pdf. Accessed 19 September 2011

Suh S, Huppes G (2002) Missing inventory estimation tool using extended input–output analysis. Int J Life Cycle Assess 7(3): 134–140

Suh S, Huppes G (2005) Methods for life cycle inventory of a product. J Clean Prod 13(7):687–697

Suh S, Nakamura S (2007) Five years in the area of input–output and hybrid LCA. Int J Life Cycle Assess 12(6):351–352

Tedford JR, Capps OJ, Havlicek JJ (1986) Adult equivalent scales once more: a developmental approach. Am J Agr Econ 68(2):322–333

Tukker A et al (2006) Environmental Impact of Products (EIPRO)—analysis of the life cycle environmental impacts related to the final consumption of the EU-25. In: Eder P, Delgado L (eds) Main Report, European Commission, DG Joint Research Centre (JRC). ftp://ftp.jrc.es/pub/EURdoc/eur22284en.pdf. Accessed 19 September 2011

Ulgiati S, Raugei M, Bargigli S (2006) Overcoming the inadequacy of single criterion approaches to life cycle assessment. Ecol Model 190(3–4):432–442

Vassallo P, Paoli C, Fabiano M (2009) Emergy required for the complete treatment of municipal wastewater. Ecol Eng 35(5):687–694

Vitousek PM, Mooney HA, Lubchenco J, Melillo JM (1997) Human domination of Earth's ecosystems. Science 277(5325):494–499

Weidema BP (2006) The integration of economic and social aspects in life cycle impact assessment. Int J Life Cycle Assess 11 (1):89–96

Weidema BP, Christiansen K, Nielsen AM, Norris GA, Notten P, Suh S, Madsen J (2005) Prioritisation within the integrated product policy. Environmental project no. 980. Danish Environmental Protection Agency, Copenhagen. http://www2.mst.dk/udgiv/publications/2005/87-7614-517-4/pdf/87-7614-518-2.pdf. Accessed 19 September 2011

Wier M, Lenzen M, Munksgaard J, Smed S (2001) Environmental effects of household consumption pattern and lifestyle. Econ Sys Res 13(3):259–274

Zhang TW, Dornfeld DA (2007) Energy use per worker-hour: a method of evaluating the contribution of labour to manufacturing energy use. In: Shozo T, Yasushi U (eds) Proceedings of the 14th CIRP Conference on Life Cycle Engineering 2007, advances in life cycle engineering for sustainable manufacturing businesses. Waseda University, Tokyo, pp 189–194